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Groundwater nitrate following installation of a vegetated riparian buffer

Toshiro Yamada [a,1], Sally D. Logsdon [b,*], Mark D. Tomer [b,2], Michael R. Burkart [c,3]

[a] National Institute of Public Health, 2-3-6, Minami, Wako, Saitama 351-0197 Japan
[b] National Soil Tilth Laboratory, 2150 Pammel Drive, Ames, IA 50011, United States
[c] Iowa State University, 352 Science I, Ames, IA 50011, United States

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Abstract

Substantial questions remain about the time required for groundwater nitrate to be reduced below 10 mg L^{-1} following establishment of vegetated riparian buffers. The objective of this study was to document changes in groundwater nitrate–nitrogen (NO_3–N) concentrations that occurred within a few years of planting a riparian buffer. In 2000 and 2001 a buffer was planted adjacent to a first-order stream in the deep loess region of western Iowa with strips of walnut and cottonwood trees, alfalfa and brome grass, and switch grass. Non-parametric statistics showed significant declines in NO_3–N concentrations in shallow groundwater following buffer establishment, especially mid 2003 and later. The dissolved oxygen generally was >5 mg L^{-1} beneath the buffer, and neither NO_3–N nor DO changed significantly under a non-buffered control area. These short-term changes in groundwater NO_3–N provide evidence that vegetated riparian buffers may yield local water-quality benefits within a few years of planting.
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1. Introduction

Vegetated riparian buffers have long been viewed as a useful tool to water-quality problems when they are located at the edge of fields and adjacent to streams. Recently, the installation of riparian buffers in the Midwest U.S. has become much more common with the National Conservation Buffer Initiative underwritten by the United States Department of Agriculture. Buffers are being installed under this initiative in anticipation of a variety of environmental improvements including water quality. Water-quality benefits that are expected include removal of 50% or more of nutrients and biocides, 60% or more of certain pathogens, and 75% or more of sediment (NRCS, 2001). Schultz et al. (1995), Correll (1996), and other authors, have reviewed and summarized the general mechanisms by which vegetated riparian buffers may trap, transform, and remove sediment, nitrate, phosphorus, and biocides in solution and adsorbed on suspended particulate matter.

* Corresponding author. Tel.: +1 515 294 8265; fax: +1 515 294 8125.
E-mail addresses: t-yamada@niph.go.jp (T. Yamada), sally.logsdon@ars.usda.gov (S.D. Logsdon), mark.tomer@ars.usda.gov (M.D. Tomer), mburkart@iastate.edu (M.R. Burkart).
[1] Tel.: +81 48 458 6156; fax: +81 48 458 6275.
[2] Tel.: +1 515 294 0213; fax: +1 515 294 8125.
[3] Tel.: +1 515 451 5657; fax: +1 515 294 6049.

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Evidence documenting reductions in groundwater nitrate concentrations attributable to recently planted vegetated riparian buffers is inconsistent. Although nitrate (NO_3–N) can be reduced in runoff, the dominant transport mechanism for a readily soluble contaminant such as NO_3–N is groundwater flux (Dillaha, et al., 1989; Haycock and Pinay, 1993; Clausen et al., 2000). Studies of buffer impacts on groundwater quality show a definite tendency toward removal of soluble nutrients, particularly NO_3–N. Beneath the Bear Creek constructed buffer in central Iowa, denitrification potential and NO_3–N loss were reported (Spears et al., 1998; Simpkins et al., 2002, although a direct connection with the multi-species buffer vegetation was not established. Riparian buffers of established forests were found to reduce groundwater NO_3–N concentrations by 48% (Snyder et al., 1998), more than 85% (Lowrance, 1992), 95% (Spruill, 2000), and almost 100% (Verchot et al., 1996) compared to non-buffered riparian areas. The scale and rates of NO_3–N losses under young riparian forests is relatively unknown, although processes similar to those beneath established forests should be occurring. Other studies clearly define reductions in NO_3–N beneath a variety of buffer vegetation and soil and groundwater conditions (Peterjohn and Correll, 1983; Nelson, et al., 1995; Devito et al., 2000; Spruill, 2000; Rosenblatt et al., 2001; Gold et al., 2001). These reductions were attributed to plant consumption and particularly denitrification in the presence of abundant organic carbon available beneath the buffers.

Clearly, NO_3–N can be removed from groundwater as it infiltrates or flows beneath riparian buffers. An understanding of water flow patterns is needed to determine the effectiveness of buffers (Correll and Weller, 1989; Hill, 1990; Gilliam, 1994; Gilliam et al., 1997; Simpkins et al., 2002), in part because of the variability in documented rates of NO_3–N removal and the time needed to reduce NO_3–N concentrations in groundwater beneath riparian buffers. Much of the variability may also be due to differences in the vegetation (Haycock and Pinay, 1993), scale of buffers (Mander et al., 1997), and landscape attributes of the buffers and sites.

The purpose of this paper is to determine if groundwater NO_3–N can be reduced rapidly following planting of a riparian buffer, within 5 years. The buffer experiment was designed to test multiple hypotheses, one of which was that groundwater NO_3–N concentrations would be decreased beneath buffer vegetation.

2. Materials and methods

2.1. Study area

The buffer study site (41.2°N, 95.6°W) in this paper is located along a first-order stream in the long-term research watersheds of the Deep Loess Research Station (DLRS) managed by the Agricultural Research Service (Karlen et al., 1999). These watersheds typify the agriculture, terrain, and hydrology of headwaters in the region. The buffer site is adjacent to a field with several decades of intensive row crop production including periods of continuous maize that requires regular nitrogen fertilizer inputs (Karlen et al., 1999). This included a 1970s experiment that almost tripled the application of nitrogen fertilizer to the adjacent field that had a long-term impact on groundwater nitrate (Tomer and Burkart, 2003). The site provided access to a first-order stream where it will be possible to monitor long-term stream responses to the buffer. Soils and subsurface materials are relatively uniform along both riparian areas of the stream along a enough distance to allow measurements in both a buffered area and one without a vegetated buffer along a stream-reach of large-enough scale to account for edge effects.

The deep loess region of western Iowa, northwest Missouri, and eastern Nebraska, also known as Major Land Resource Area 107 (USDA, 1981), is underlain by loess that is generally 20 m thick, but can be as much as 60 m thick (Prior, 1991). Loess thickness in the watershed studied does not exceed 21.5 m. Materials eroded from slopes and uplands were locally redeposited during Wisconsinan, Holocene and Recent periods forming slopewash, colluvium, and alluvium on the toe slopes and in the valleys of even first-order streams such as those studied at this buffer site (Bettis, 1990). Under the loess layer is glacial till (pre-Illinoisan) and a layer of clay (Yarmouth Clay) that retards downward water movement. Soils developed in the deep loess are dominantly mapped as Monona (Typic Hapludolls), and Ida (Typic Udorthents) soils. Napier and Kennebec (Cumulic Hapludolls) soils are developed on the colluvium and alluvium in the riparian areas (Soil Survey Staff, 1994).

The column of loess, colluvium, and alluvium provides a medium for relatively continuous storage and flux of water that supports perennial baseflow to streams. The hydraulic conductivity of the loess, paleosols, and the colluvium and alluvium derived from loess are much greater (10^{-5} m s^{-1}) than that of the underlying Yarmouth Clay with a K_s of $<10^{-11}$ m s^{-1} and the (pre-Illinoisan till with a K_s of 10^{-6} m s^{-1}). This difference in hydraulic conductivity creates a zone of saturation in the loess above the Yarmouth Clay or pre-Illinoisan till that provides baseflow to the streams in the watershed.

2.2. Vegetated riparian buffer

A three-part riparian buffer, 183 m long, was planted in the fall of 2000 and spring of 2001 on the west bank of a first-order stream. The buffer was designed with three parts: trees in a 15 m strip adjacent to the stream edge; an approximately 5 m strip of alfalfa (*Medicago sativa* L.)and brome grass (*Bromus inermis* Leyss) adjacent to the trees; and an approximately 5 m strip of switch grass (*Panicum virgatum* L.) between the crop edge and the alfalfa/brome. Tree species were selected to provide a combination of long-term, high-value trees (walnut, *Juglans nigra* L.) surrounded by rapidly growing trees (cottonwood, *Populus deltoides* Bartr.) that could nurse the growth of the walnuts and be harvested on approximately five-year intervals without replanting. Planting was initiated with grass, alfalfa and walnut seeds on October 10, 2000. Alfalfa and switch grass were planted at a rate of 11 kg ha^{-1} and brome at a rate of 18 kg ha^{-1}. Timothy (*Phleum pratense* L.) was underseeded in the area to be occupied by trees for weed suppression. Approximately 1200 walnuts were planted in a line 2 m equidistant from two pairs of cottonwood-tree lines at intervals of 15 cm and covered with a strip of poultry screen after losing many walnuts to local wildlife during the initial planting. On April 13, 2001 244 rooted cottonwood clones approximately 20 cm long were planted at an interval of 3 m in two pairs of rows 3 m apart. Beyond the buffer vegetation, the watershed continued under corn (*Zea mays* L.)–soybean [*Glycine max* (L.) Merr] rotation. No new buffer was planted on the east bank, to provide the reference conditions to compare any changes that may occur beneath the buffer; however, a 5–7 m wide strip of smooth brome grass remained, with corn–soybean rotation beyond the smooth brome grass.

The greatest growth of cottonwood trees was in 2003 based on annual measurements. During this year, the median values for base diameter, tallest stem height, and diameter at 1.22 m all showed the greatest increase and full canopy closure was reached in most locations (M. Kelly, personal communication, 2006). Cottonwood shoot growth was 0.2, 0.5, 2.7, and 9.0 kg m^{-2} for end-of-season 2001 through 2004 (Kelly et al., 2007). These data lead to the assumption that 2003 represented the first substantial water use and nutrient uptake by the trees in the buffer. Consequently, March 1, 2003, the start of the growing season, was selected to represent the beginning of the period that would exhibit any measurable groundwater response to the buffer. Similarly, Tomer et al. (in press) stated “this buffer's vegetation first became established enough to effectively trap sediment near the end of the 2002 growing season, in an evaluation of sediment and phosphorous accumulation in this buffer's switchgrass vegetation zone.”

2.3. Piezometers

Two piezometer nests were installed in 1996 (1T and 1V, Fig. 1) as part of a watershed-scale groundwater analysis. In 1999 additional piezometer nests were installed in the riparian area to measure hydraulic and water chemistry variables before and after establishing the vegetated buffer. Three transects of nests were

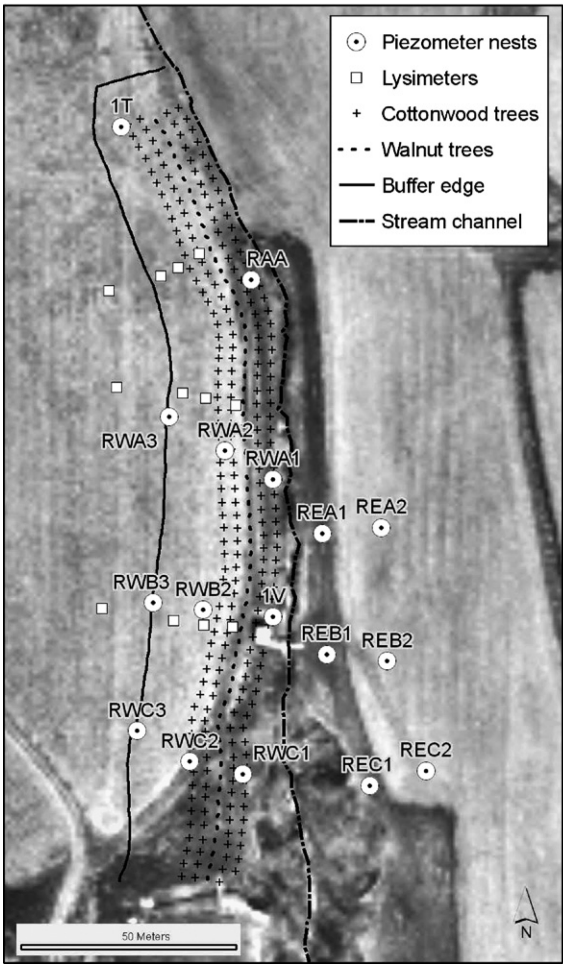

Fig. 1. Location of piezometer nests, lysimeters, and riparian buffer along a first-order stream.

installed along hypothesized groundwater flow paths on the west side of the stream (designated by RW) where the buffer was later installed and the east side (designated by RE) where a narrow (<5 m) brome grass strip separated the crop from the stream (Fig. 1). Nests were further identified by the transect position (A most northern and C most southern) and proximity to the stream (1—closest to stream; 2—mid buffer; and 3—field edge). The piezometer nests were installed at the lines of vegetation change (cottonwood to smooth brome, smooth brome to switchgrass, switchgrass to crop). Nest RAA was added north of the A transect to provide information intermediate between the transect and nest 1T located near the northeast corner of the buffer. Each individual piezometer is specifically identified with a number representing the depth to the top of the screen in meters. Piezometers 3 to 18 m deep were constructed of 50 mm i.d. PVC with 0.6 m screens. The x, y, and z coordinates locating the head measurement point for each piezometer were determined using total station surveys and tied to local benchmarks. More details on piezometer/well installation can be found in Tomer and Burkart (2003).

2.4. Hydrologic and nitrate measurements

Head measurements were made weekly using an electric line. The hydraulic conductivity (K_s) of the saturated zone was measured once, following well development, by conducting slug tests in each piezometer. A solid PVC rod was lowered and later raised to conduct slug-down and slug-up tests. Hydraulic head was measured during each slug test using a pressure transducer and data logging system. The slug test data were analyzed using the Hvorslev (1951) method. At each piezometer, the reported K_s is the mean of the values measured by rising and falling head tests. Monthly water levels were measured in each piezometer using an electric sensor.

Upon completion, the piezometers were sampled monthly and analyzed for NO_3–N concentrations. NO_3–N was analyzed using flow-through injection analysis of nitrite and NO_3–N reduced in a copperized Cd column by the National Soil Tilth Laboratory in Ames, IA. An auto-analyzer technique was used with a 0.3 mg L^{-1} quantitation limit described by Hatfield et al. (1999). This method provides the total NO_3–N plus NO_2–N without providing quantitative results for each species. At least 1.5 water-column volumes were purged from each piezometer before sampling. Recharge to a few piezometers east of the stream in the non-buffered area was too slow to allow such purging, and sampling of these installations was done either less frequently (e.g., twice per year) or suspended. During sampling dissolved oxygen (DO) measurements were made in the field using a flow-through chamber and a colorimetric method that utilized individual ampoules manufactured by CHEMets (www.chemitrics.com).

2.5. Soil moisture measurements

The purpose of the neutron soil water measurements was to characterize the unsaturated zone soil water content, based on calibration specific to the loess at the site. Neutron probe access tubes were installed in 2000 near the piezometer nests. The access tubes were 50 mm diameter steel tubing and were installed with a hydraulic Giddings probe to 2.5 m if possible. Bentonite was used to seal around the top 0.2 m to prevent water flow down the annulus of the tube. Neutron probe measurements were done weekly or biweekly during the growing season and were read at 0.2 m increments starting at 0.3 m. A volumetric sampler (Pikul and Allmaras, 1986) was used to collect surface samples (0–10, 10–20, and 20–30 cm) for water content at the same time the neutron probe measurements were done. The data were summarized to 1.1 m depth because at a few sites and times, the water table was as shallow as 1.1 m. Map coordinates and elevations of neutron probe tubes were determined by a differential global positioning system (GPS) survey with relative errors of 0.01 m in horizontal and vertical dimensions.

2.6. Vadose zone nitrate

Suction lysimeters were installed along three transects located between piezometer nest sites (Fig. 1), each transect having lysimeter installations in each of the vegetated zones of the buffer and in the adjacent crop. A pair of lysimeters were installed at 1.2 m depth at each transect location. These were sampled monthly or bimonthly, and the water extracted from each pair of lysimeters was combined. On some sampling dates, conditions were too dry to provide enough sample to analyze, but combining the paired samples reduced the frequency of this. Samples were analyzed for NO_3–N concentrations using an auto-analyzer technique with a 0.3 mg L^{-1} quantification limit described by Hatfield et al. (1999).

2.7. Statistical analyses

Non-parametric statistics were used to analyze NO_3–N and DO trends in samples from piezometers beneath the planted buffer and in the non-buffered area east of the stream. Measurements beneath the east side of the stream were designed to be measurements under control conditions in similar soils, geologic materials, and farming conditions where no vegetated riparian buffer was planted. Non-parametric statistical methods are particularly useful for analyses of censored or truncated data like the NO_3–N data gathered in this study that includes "less than" values. Data on NO_3–N concentrations from one location (RWA1-12) and all DO data had non-normal distributions, which was shown by the Kolmogorov–Smirnov test for a normal distribution ($p<0.05$). Non-parametric statistics are useful for examining non-normally distributed data and provide the opportunity to include measurements below an analytical limit, such as NO_3–N, for which a specific numerical value cannot be precisely defined. To determine any changes in NO_3–N and DO associated with the buffer, the data were divided into two periods separated by March 1 2003. This is the date after which the rate of tree growth in the buffer was substantial and

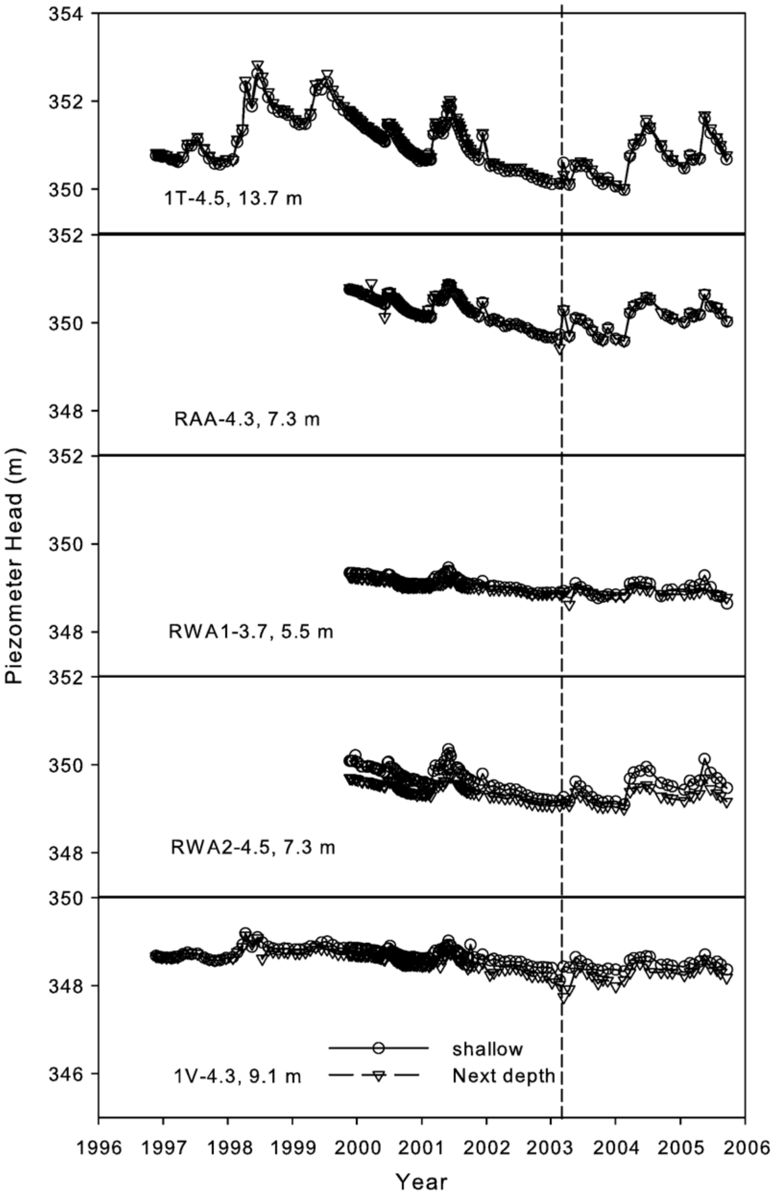

Fig. 2. Hydraulic heads in selected piezometer nests located in the planted vegetated riparian buffer. Numbers adjacent to symbols denote the depth in m of individual piezometers at each site.

the date hypothesized to represent the beginning of measurable groundwater responses to the buffer. Trend tests were separately applied to data from the two periods. The Mann–Kendall test was used for analyzing long-term trends in NO_3–N and DO. This non-parametric test is useful to detect monotonic trends and it accommodates non-normal data distribution (Hirsch et al., 1982). Step-trend tests were also conducted to detect any differences between NO_3–N concentrations for the two periods using the Wilcoxon–Mann–Whitney rank-sum test (Helsel and Hirsch, 1992).

3. Results and discussion

3.1. Groundwater flow

The saturated zone in the watershed dominantly occurs within the loess upslope of the buffer and in alluvium beneath the alluvium. The loess thickness declines downslope from a maximum of 21.5 m at the divide to the line where it has been truncated by erosion and replaced by colluvial and alluvial deposits. The geometric mean K_s of the loess was measured to be 1×10^{-5} m s^{-1} and that of the was 4.1×10^{-6} m s^{-1}. Given the similar K_s of the loess, colluvium and alluvium and their common origin discussed above, they are interpreted to be a continuous flow system separated from deeper aquifers by either the Yarmouth Clay or the pre-Illinoisan Till. An analysis of the general groundwater flow in the watershed shows a consistent gradient from higher positions on the landscape toward the stream with consistent upward gradients in the area beneath the buffer (Tomer and Burkart, 2003). Changes in management in fields adjacent to the buffer are unlikely to have influenced the NO^3–N available in the groundwater beneath the buffer because multiple lines of evidence showed that it takes several decades for groundwater to travel from the divide to the stream (Tomer and Burkart, 2003).

3.2. Hydraulic head trends

The long-term hydraulic head trend in all piezometers was generally negative over the complete period of record. Upward gradients of between 0.02 and 0.08 m m^{-1} were temporally consistent between the upper two piezometers at most sites as represented in Fig. 2. This upward gradient seen both before and after the buffer installation shows that the source of groundwater to the buffer zone was consistently from groundwater flowing beneath the buffer. Some seasonal recovery in hydraulic heads was observed during late winter and spring of most years, particularly 2003–2005 (Fig. 2) in response to more consistent summer precipitation than in previous years. Patterns of precipitation, soil moisture, and hydraulic heads (Fig. 3) were closely linked. The crop demands for water in the unsaturated zone is substantial so that only sustained periods of precipitation or large storm precipitation during the growing season (May through August) would provide a surplus to recharge groundwater in that period. Typically, during the growing season (1/3 of the year) the watershed receives an average of 48% of the mean annual precipitation (794 mm) (Logsdon et al., 1999). During 2001 to 2003 the annual precipitation was from 609 to 748 mm and an average of only 40% of the total fell during the growing season with 66% in 2004. Total annual as well as seasonal precipitation were below average except in 2004. The larger annual and seasonal precipitation in 2004 was due to more and larger summer thunderstorms than other years. Lack of intense thunderstorms through 2003 prevented significant recharge during the growing season. Through this variation in seasonal recharge, there is clear evidence that the upward gradient within the buffer persists (Fig 2). The long-term trends showed a consistent decline both beneath the buffer as well as the non-buffered riparian zone east of the stream. Variations in the scale of seasonal trends reflect different combinations of the K_s of the material in which the piezometer was screened, the variability in recharge area, and the direction and magnitude of vertical flux.

Intense water uptake can occur at the capillary fringe (Reicosky et al., 1972). Also appreciable quantities of water can move up from the water table into the root zone, particularly in silt-loam soils (Van Bavel et al., 1968; Allmaras et al., 1975; Van Bavel and Ahmed, 1976; Maraux and Lafolie, 1998), replenishing soil water lost to plant uptake. Groundwater flow from upslope in the watershed continued throughout the study period (Tomer and Burkart, 2003) which replenished the groundwater beneath the buffer. These combinations of factors also made it difficult to document increased water uptake in the buffer.

3.3. Dissolved oxygen

Non-parametric regression showed that dissolved oxygen (DO) increased in most locations beneath the buffer, while no changes in DO occurred beneath the non-buffered area. Dissolved oxygen concentrations near the top of the water table (as indicated from piezometer measurements) were generally greater than 1.0 mg L^{-1} (Tables 1 and 2). Although several sites had concentrations less than 1.0 mg L^{-1}, these lesser concentrations occurred early in the period of record during the first year following piezometer installation.

Statistically significant trends (Mann–Kendall test) in DO from piezometers beneath the buffer (Table 1) were almost exclusively positive (increasing DO) during the entire period of record (1997–2005). Trends in DO from piezometers beneath the non-buffered area east of the stream (Table 2) were either not significant or significantly negative. Significant positive trends were also calculated for the pre-buffer period beneath the buffer area and, with one exception (1T), the DO trend in all shallow piezometers changed from significantly positive to no significant trend after establishment of the buffer. The trend seen in 1T may result from improved vertical recharge in an area where vehicle traffic had compacted surface soils prior to establishment of the buffer. The shift from a significant increase in DO before establishing the buffer to no significant trend after establishment may represent the effect of reduced vertical groundwater recharge through the buffer or increased scavenging of oxygen associated with increased root carbon from perennial vegetation. Reduced groundwater recharge may have resulted from increased demand by the perennial buffer vegetation compared to annual crops that occupied the site earlier. There were no sites where the DO concentration was reduced to

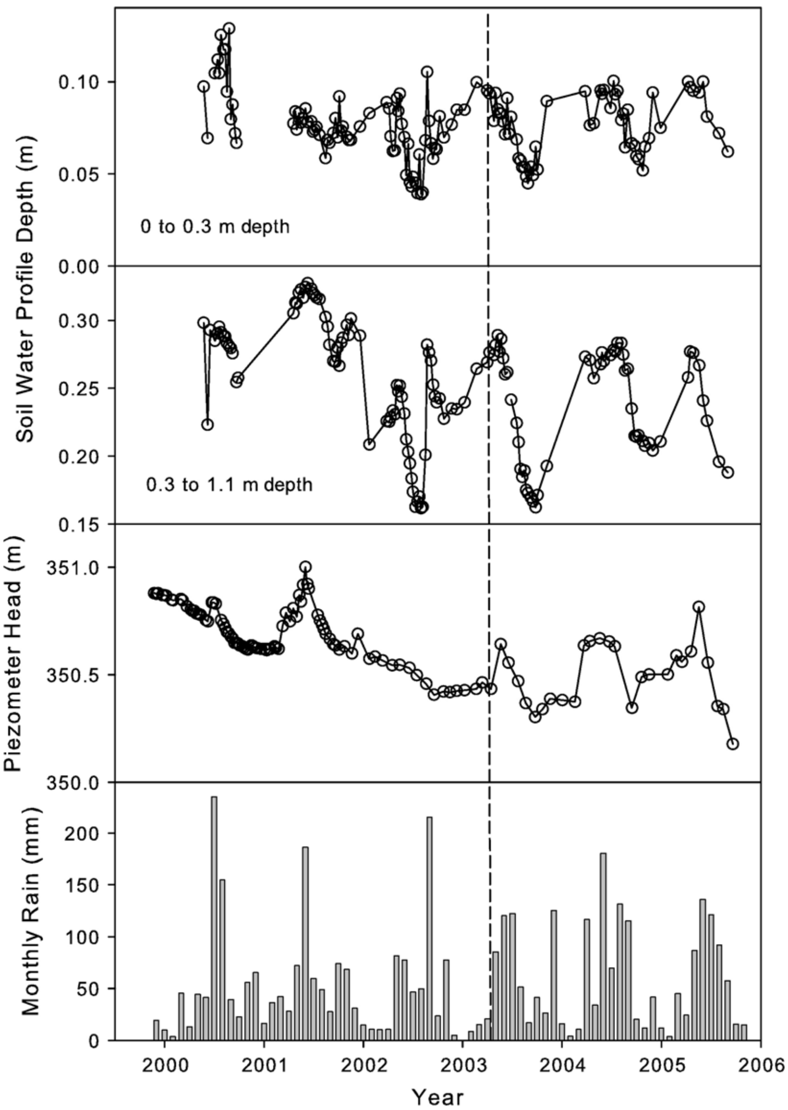

Fig. 3. Soil moisture, hydraulic head and precipitation at a representative site (RAA) in the vegetated riparian buffer. (a) Soil water content from neutron probe data with depth light shades represent drier conditions than dark shades; (b) Hydraulic head in shallowest piezometer; (c) Monthly precipitation in watershed.

concentrations less than 1.0 mg L^{-1}, a concentration below which denitrification is generally considered to be active (Tiedje, 1988).

3.4. Nitrate concentrations

Non-parametric regression showed that nitrate (NO_3–N) trends in samples from the eastern non-buffered riparian zone were similar before and after March 2003 using the Mann–Kendall test (Fig. 4). The piezometers shown are those beneath a field-border of brome grass that existed east of the stream throughout the sampling period. This similarity in temporal NO_3–N trends provide the control conditions upon which those beneath the buffer are evaluated. Piezometers on the east side produced generally lower NO_3–N concentrations than those initially measured on the buffered side west of the stream (Figs. 4 and 5). The larger concentrations on the west side may be the leading edge of larger NO_3–N concentrations shown to be moving toward the riparian area as a result of an experiment between 1969 and 1974 (Tomer and Burkart, 2003) that over-fertilized this part of the watershed.

Some of the piezometers (RWC-2, RWC-1, and RWB-2) had few detects of NO_3–N at the shallowest depth (Table 3), but higher concentrations at deeper depths (not shown), and differences in piezometric heads indicating upward gradients at these sites. These sites had no decrease in NO_3–N during the study within shallow groundwater because they started with none. This was not the pattern for the rest of the piezometer sites.

Application of the Mann–Kendall test showed NO_3–N concentrations beneath the buffer had significant reductions or no significant trend in several locations after March 2003 as compared to concentrations before buffer establishment (Table 3). Positive trends prior to establishing the buffer may reflect the lateral flux of large NO_3–N concentrations into this part of the groundwater system. Tomer and Burkart (2003) reported on the slow discharge of excess N application, average of 446 kg–N ha^{-1} yr^{-1} (398 lb-N ac^{-1} yr^{-1}) between 1969 and 1974, which is still affecting parts of the groundwater flow system. Also, a shift from conventional tillage to no-till in 1996 (Karlen et al., 1999) may have increased infiltration and NO_3–N leaching as suggested by Randall and Goss (2001) may have had an effect on the NO_3–N concentrations observed in shallow groundwater. Significant reductions in NO_3–N concentrations were observed in spite of the changes in land management upgradient to the buffer (Tomer and Burkart, 2003) that continue to supply additional groundwater NO_3–N to the buffer area.

Table 2
Mann–Kendall test results for dissolved oxygen concentrations from the shallowest piezometers in each nest beneath the non-buffered area (non-parametric regression)

Site	Piezometer depth (m)	Range mg L^{-1}	Period of record		
			104/1997–09/2005	04/1997–02/2003	03/2003–09/2005
REA1	4.0	0.05–6	(+)	(+)	(−)
REA2	3.7	5–9	−	(−)	(+)
REB1	4.3	1–6	(+)	(−)	(+)
REB2	3.4	5–8	−	(−)	(+)
REC1	4.9	5–7	(−)	(−)	(+)
REC2	4.3	6–10	−	−	(+)

+ = significant increase ($p<0.05$); − = significant decrease ($p<0.05$); (+) = non-statistically significant increase; (−) = non-statistically significant decrease; (0) = no trend.

Table 1
Mann–Kendall test results for dissolved oxygen concentrations from the shallowest piezometers in each nest beneath the buffer (non-parametric regression)

Buffer vegetation	Site	Piezometer depth (m)	Range mg L^{-1}	Period of record		
				04/1997–09/2005	04/1997–02/2003	3/2003–09/2005
Edge: crop and switch grass	1T	3.0	1–10	+	+	+
	RWA3	4.6	0.2–6	+	+	(−)
	RWB3	4.3	4–6	+	+	(+)
	RWC3	4.6	1–9	(−)	(+)	(0)
Edge: brome grass/alfalfa and trees	RWA2	4.6	0.3–6	+	+	
	RWB2	3.4	0.3–6	(−)	(+)	(+)
	RWC2	4.6	0.05–6	+	+	(+)
Trees	RAA	4.3	1–5	+	(+)	(+)
	RWA1	3.7	0.3–6	+	+	(−)
	1V	4.6	1–5	+	+	(−)
	RWC1	4.6	0.01–6	+	+	(+)

+ = Significant increase ($p<0.05$); − = significant decrease ($p<0.05$); (+) = non-statistically significant increase; (−) = non-statistically significant decrease; (0) = no trend.

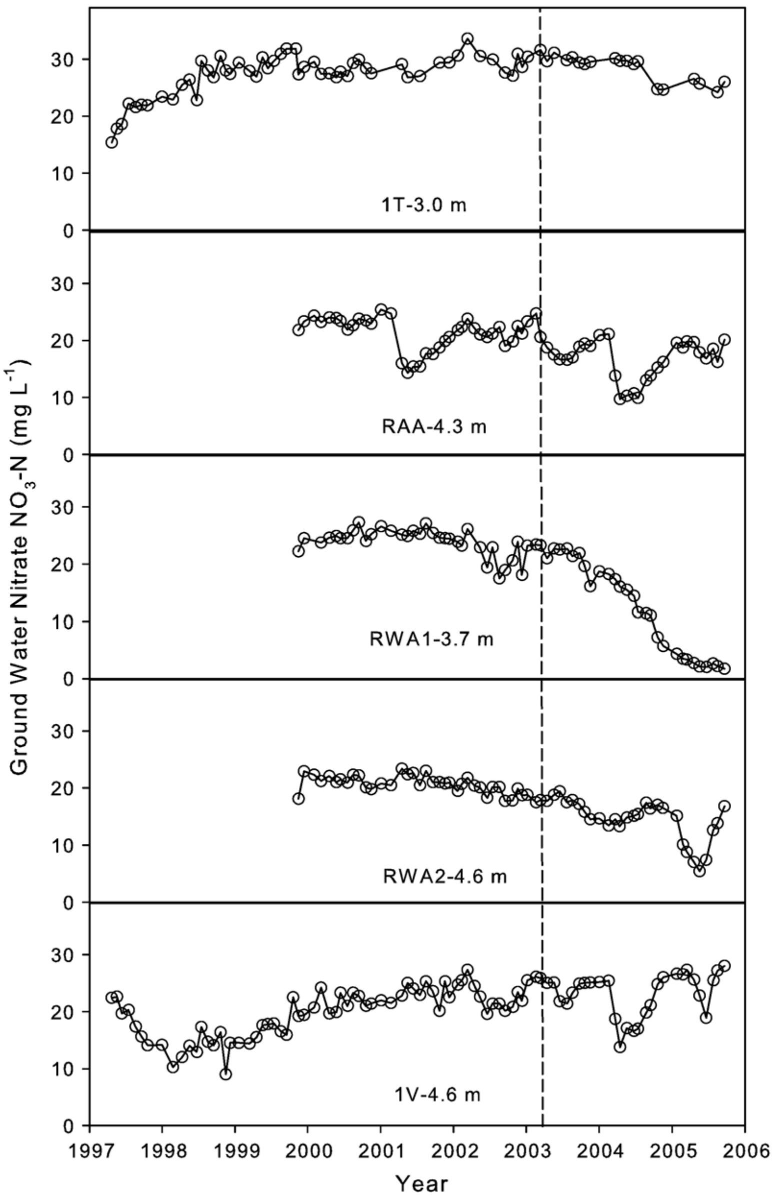

Fig. 4. Nitrate concentrations in the shallowest piezometers beneath the non-buffered area east of the creek.

In nest 1T, the period after the trees had grown had a significant negative trend (Table 3) representing a change to conditions under which NO_3–N was consumed by plants or microbes or denitrified, even though this shift in trend is not as apparent when NO_3–N concentrations are plotted with time (Fig. 5). In 1V, the positive trend was followed by no trend similarly interpreted as a change to conditions under which NO_3–N was consumed or denitrified after establishment of the buffer vegetation. Significant negative trends prior to the buffer period (RWA3, RWA2, and RWA1) were followed be either a non-significant positive trend (RWA3) or a significant negative trend (RWA2 and RWA1). In both RWA2 and RWA1, the negative trends have steeper slopes following buffer development (Fig. 5). The change in NO_3–N trend is most obvious in RWA1 (Fig. 5) where a small but significant decline in NO_3–N prior to buffer establishment shifts to a steep decline that ends with concentrations equal to or less than the quantitation limit.

Results of the Wilcoxon rank-sum test (*U*-test) showed more clearly that significant reductions in NO_3–N concentrations occurred after March 2003 (Table 4) than did the Mann–Kendall test results. The rank-sum test

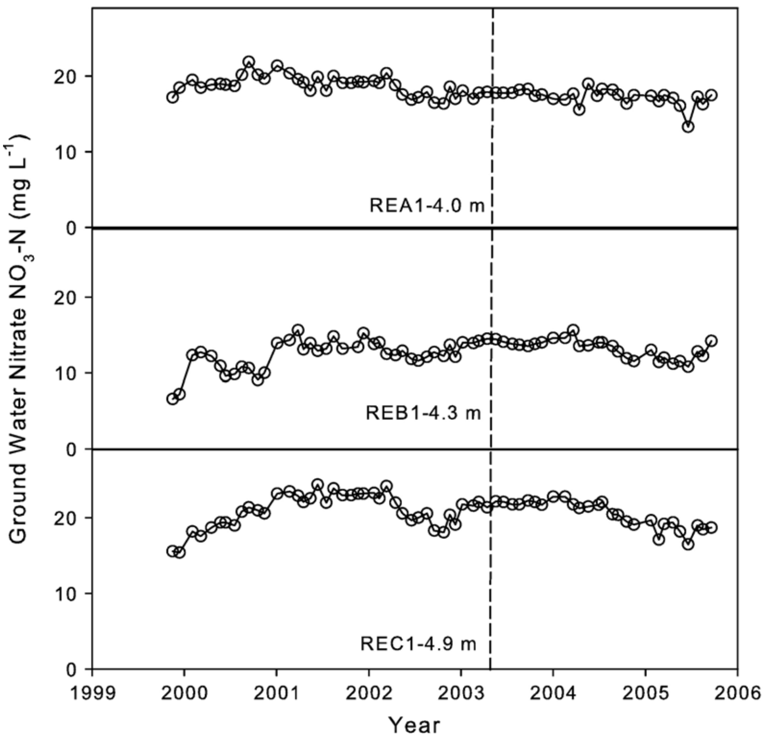

Fig. 5. Nitrate concentrations from the shallowest piezometers in representative nests beneath the buffer.

showed significant reductions in NO_3–N concentrations in all of the shallowest piezometers in nests directly under the trees as well as those nests at the boundary between the brome grass/alfalfa and the trees. Declines in NO_3–N concentrations following the period of substantial buffer growth are interpreted to result from uptake by roots of perennial plants established in the buffer, particularly those of the trees. Increases in denitrification rates are not likely to be the major factor responsible for declines in NO_3–N this early in the development of the buffer because there is no evidence of substantial reductions in DO in any of the piezometers.

Table 3
Mann–Kendall test results for nitrate from the shallowest piezometers in each nest beneath the buffer, and for the lysimeters (non-parametric regression)

Buffer vegetation	Site	Range, mg L^{-1}	Piezometer depth (m)	Period of record		Lysimeter
				04/1997–02/2003	03/2003–09/2005	10/2002–9/2005
Edge of crop and switch grass	1T	17–32	3.0	+	−	+
	RWA3	1–11	4.6	−	(+)	(+)
	RWB3	26–30	4.3	(−)	−	(−)
	RWC3	23–29	4.6	(−)	(−)	+
Edge: brome grass/alfalfa and trees	RWA2	5–23	4.6	−	−	(+)
	RWB2	0	3.4	(−)	(+)	(+)
	RWC2	0–2	4.6	(−)	(−)	(+)
Trees	RAA	10–24	4.3	(−)	(+)	(−)
	RWA1	2–27	3.7	−	−	−
	1V	14–28	4.6	+	(+)	−
	RWC1	0–2	4.6	(0)	(−)	(−)

+ = significant increase ($p<0.01$); − = significant decrease ($p<0.01$); (+) = non-statistically significant increase; (−) = non-statistically significant decrease; (0) = no trend.
[†]Crop, SG(switchgrass), SB(smooth brome), CW(cottonwood) are in the north(N), mid (M), or south (S) position (Fig. 1).

Table 4
Wilcoxon rank-sum tests (*U*-test) results for nitrate concentrations before and after March 1, 2003 from the shallowest piezometers in each nest beneath the buffer

Buffer vegetation	Site	Piezometer depth (m)	*p* value
Edge of crop and switch grass	1T	3.0	$p=0.26$
	RWA3	4.6	$p=0.29$
	RWB3	4.3	$p=0.88$
	RWC3	4.6	$p=0.81$
Edge: brome grass/alfalfa and trees	RWA2	4.6	
	RWB2	3.4	$*p<0.001$
	RWC2	4.6	$*p<0.01$
Trees	RAA	4.3	**
	RWA1	3.7	
	1V	4.6	$*p<0.001$
	RWC1	4.6	$*p<0.001$

* = significantly different ($p<0.01$); ** nitrate concentrations less than the quantitation limit (<0.03 mg l^{-1}) during the period of record.

3.5. Lysimeter nitrate data

Nitrate concentrations increased significantly over time for one crop site and non-significantly for the other sites (Fig. 6), probably due to continual fertilizer input beyond crop need. Nitrate concentrations in lysimeters beneath the switchgrass increased, perhaps because of N-recycling from plants litter. Also the switchgrass slowed runoff from upslope and increased infiltration. Zones of sediment accumulation were apparent at the edge of the crop and switchgrass strip. In contrast, cotttonwood trees had been thinned periodically, and average NO_3–N concentrations from the lysimeters decreased beneath them. For nearly all measurement dates, NO_3–N concentrations in lysimeters (Fig. 6) were significantly less under riparian vegetation (5 mg L^{-1}) than under the adjacent crop (24 mg L^{-1}), because no fertilizer was applied to the buffer, and because of additional uptake of N by buffer plants. Reduced vadose zone NO_3–N concentrations under the riparian vegetation reduced downward NO_3–N leaching to the saturated zone. Except under corn/soybean rotation, lysimeter NO_3–N concentrations were less than in the underlying shallow groundwater (Table 3, Fig. 6), neglecting the piezometers with non-detectable NO_3–N throughout the study. The water table NO_3–N in the buffer was therefore primarily contributed from lateral or upward saturated flow rather than from downward percolation through the soil.

4. Summary and conclusions

Significant reductions in groundwater NO_3–N concentrations for some sites were shown to be associated with the early stages of development of a planted riparian buffer when compared to a non-buffered area. Substantial growth in trees planted for the buffer has been reported for only three years during which NO_3–N changes have occurred. The buffer, developed along a first-order stream in the deep loess region of western Iowa, included a strip of cottonwood and walnut trees adjacent to the stream, a strip of brome grass/alfalfa, and a strip of switch grass adjacent to the cropped field.

Non-parametric statistical tests provided evidence of declines in the trends of NO_3–N concentrations in shallow saturated and unsaturated groundwater following buffer establishment, especially for certain piezometer nests. These tests also showed trends in reductions

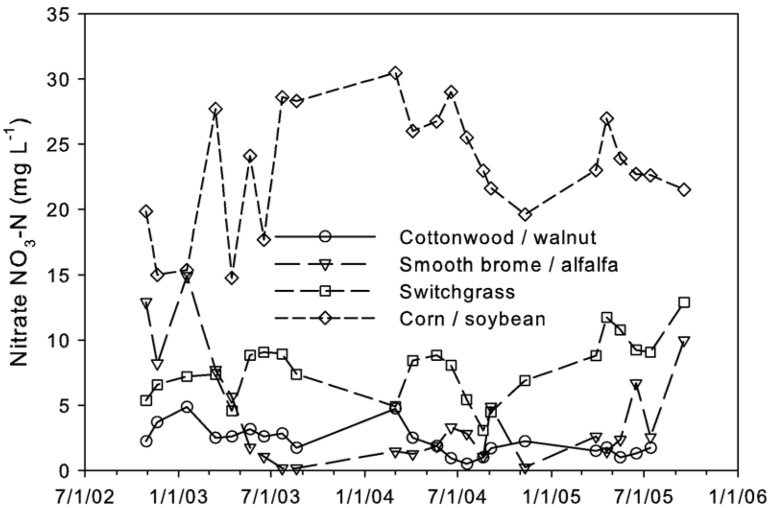

Fig. 6. Mean soil lysimeter nitrate–N from mid 2001 to October 2005 for riparian vegetation and adjacent crop.

in NO_3–N concentrations when data prior to and following substantial tree growth were compared. More substantial decreases were seen for soil NO_3–N concentration from lysimeters under riparian vegetation than from lysimeters under corn/soybean rotation.

Concentrations of DO generally remain in excess of 5 mg L^{-1} in all shallow piezometers beneath the buffer. However, non-parametric tests showed significant reductions in the trend of increased DO concentrations in samples from these piezometers. Consequently, the decline in groundwater NO_3–N is attributed primarily to plant uptake, although denitrification might have been a minor contribution, especially in the piezometer sites with few detects of NO_3–N.

Results of such short-term changes in groundwater NO_3–N nitrate provide evidence that vegetated riparian buffers may yield water-quality benefits in groundwater within a few years. Additional evidence in other settings will be required to provide support for general concepts about general groundwater responses to this conservation practice.

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